CHAPTER 17

BIOLOGICAL RESPONSE SIGNATURES AND THE AREA OF DEGRADATION VALUE: NEW TOOLS FOR INTERPRETING MULTIMETRIC DATA

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1.0 INTRODUCTION

Biological criteria are used primarily as an environmental assessment tool and are intended to reflect the relative health and well-being of the resident aquatic community. The most appropriate use of biological criteria, and the attendant ambient bioassessment techniques, is in assessing the relative condition of the aquatic resource with regard to temporal and spatial variables and influences. Many states use biological assessment as the principal approach for determining the extent of impairments of the aquatic resource, and water quality in general. In Ohio, the appropriate role of biological criteria is defined by the Ohio Water Quality Standards as the principal arbiter of aquatic life use goal attainment/non-attainment (Yoder and Rankin, Chapter 9).

Interpretation of the biological results, and resulting use attainment status, is best done on a longitudinal basis (e.g., river reach). For example, a site exhibiting partial attainment in the midst of nonattaining sites that extend for some distance both upstream and downstream would be viewed as an exception, with the assessment of widespread nonattainment prevailing. Conversely, the interpretation of this marginal result would have been quite different if the attainment status of the other sites was full. Thus, the significance, severity, and spatial extent of any observed impairment is as important as determining the attainment status. Typically, the results are portrayed as a two-dimensional graph in an upstream to downstream format and interpreted visually. In the case of Ohio, longitudinal changes are compared to the ecoregional biocriteria (Yoder and Rankin, Chapter 9). Recently, a quantitative estimate of the degree of departure from a biocriterion along a longitudinal continuum (Area Of Degradation Value), was developed to enhance the use and interpretation of biological community assessments.

Few dispute the value of biological communities to demonstrate impairment due to any number of stressors in the environment. However, the capability to use the resultant community data and information to discriminate between different stressors is frequently questioned (USEPA 1985; Suter 1993). This diagnostic capability has not been demonstrated on a widespread basis although specific examples do exist (Eagleson et al. 1990). Discernable patterns in the response of aquatic community attributes were first described using Ohio EPA data from more than 250 sites from 25 different streams and rivers and were termed *Biological Response Signatures* (Yoder 1991). This chapter presents the technical basis, with examples, for using the Area of Degradation Value and Biological Response Signatures to further interpret multimetric data for water resource management programs.

2.0 LONGITUDINAL ASSESSMENT OF AQUATIC COMMUNITY PERFORMANCE: THE AREA OF DEGRADATION VALUE

The Area of Degradation Value (ADV; Rankin and Yoder 1992) was developed to measure and quantify the longitudinal extent and magnitude of aquatic life impairment based on biological monitoring results. Thus far the ADV has been used by Ohio EPA to (1) establish priorities for wastewater treatment plant construction funding, (2) assess trends over time for contiguous stream and river segments to document the effectiveness of pollution controls, and (3) to establish priorities for point source management (e.g., development of 303[d] list) as part of an overall risk assessment to prioritize limited administrative resources. Potential uses include the assessment of penalties to polluters on the basis of actual instream damage to aquatic communities and as a component of Natural Resource Damage Assessments.

ADVs are based on measures generated from longitudinal plots of the biological indices and biological criteria as described by Yoder and Rankin (Chapter 9). The length or extent of degradation is defined simply as the distance over which the applicable index is less than the biocriterion or other benchmark value applied to that stream (Figure 1). Magnitude refers to the vertical departure from the applicable biocriterion or other benchmark value. The total ADV is the area between the criteria value and the actual data values (shaded areas in Figure 1). The computational formula for the ADV value is (using the Ohio modified version of the Index of Biotic Integrity, as an example):

$$\mathrm{ADV} = \Sigma\ [(\mathrm{pIBIa} + \mathrm{pIBIb}) - (\mathrm{aIBIa} + \mathrm{aIBIb})]\ (\mathrm{RMa} - \mathrm{RMb}),\ \text{for } a = 1 \text{ to } n$$

where: pIBIa = potential IBI at river mile a
pIBIb = potential IBI at river mile b
aIBIa = actual IBI at river mile a
aIBIb = actual IBI at river mile b
RMa = upstream river mile
RMb = downstream river mile
n = number of sampling sites

The ADV equation assumes that the average of two contiguous sampling sites accurately integrates the aquatic community status for the distance between the points. This is supported by numerous examples in rivers and streams throughout Ohio. Sampling sites are also close enough to allow meaningful changes to be tracked along a longitudinal continuum. Aquatic communities gradually recover with increased distance downstream from impacts, although the pattern may vary according to the type of impact and discharge. The connection of the sampling results produces the lines drawn in Figure 1 and should yield a sufficiently accurate representation of real changes in community performance and quantifiable departures from ecoregional biological criteria. Sampling sites are spaced more closely as the complexity of the setting increases in order to enhance this longitudinal resolution.

Figure 1 also demonstrates how different types of impacts can be layered together in a segment. The darker shading represents the impact from a point source discharge, the lighter from a nonpoint source. The effect of the point source is immediate and severe with recovery occurring with increased distance downstream. Nonpoint source impacts are more diffuse and are spread more evenly throughout a segment. In this case the true magnitude and extent of the nonpoint source impact will not be known until it is "unmasked" by the reductions of the point source impact. If the two impact types occur together in the same stream segment as portrayed in Figure 1, then the relative difference between each type can be visualized. If the point source impact(s) are fully abated with little attention given to the nonpoint sources then the recovery will be limited. Conversely, there will be little benefit to abating the nonpoint source impacts in the absence of point source controls.

2.1 Case Examples

One of the best proven uses for biosurvey data is for spatial and temporal trend analysis. Unlike chemical parameters, multimetric biological indices integrate chemical, biological, and physical impacts to aquatic systems and portray both condition and status in terms of designated use attainment/nonattainment

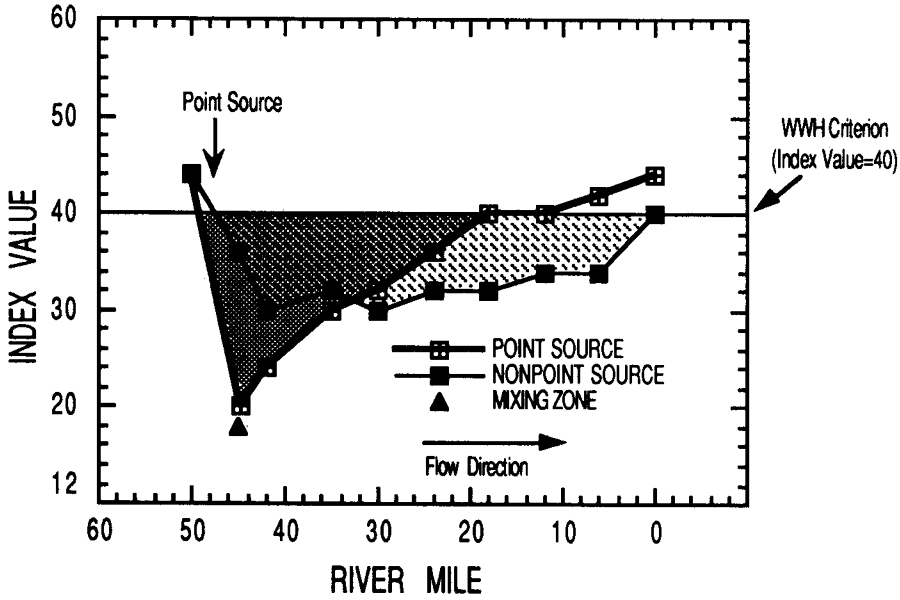


Figure 1. Portrayal of the Area of Degradation Value (ADV) concept used to estimate and quantify the severity of aquatic life impairment based on biosurvey data and ecoregional biocriteria. Examples include pre- and post upgrade of a point source (open squares) and the pattern expected for a nonpoint source impact (solid squares).

in direct terms. Frequency and duration considerations, which for chemical monitoring require large numbers of samples to adequately account for variation, are integrated by the resident aquatic life in the receiving water. Figures 2 through 6 portray biosurvey results from two Ohio rivers, each with a history of serious, but very different water pollution problems — the Scioto River and the Ottawa River. Each have been sampled over multiple years which is a prerequisite for trend analysis. The ADV statistics for these two rivers are summarized in Table 1. In both of these areas data is available before and after the July 1988 compliance deadline for municipal wastewater treatment plants (WWTPs) to meet water quality-based effluent limitations under EPA's National Municipal Policy. For most WWTPs throughout Ohio significant reductions in the loadings of pollutants such as oxygen demanding wastes, suspended solids, ammonia, nutrients, and in some instances heavy metals and other toxics has occurred. While overall loadings of conventional pollutants have declined in both rivers, differing biological results were evident in each. The Ottawa River has some significant and residual toxic chemical problems remaining, which have precluded the magnitude of improvements observed in the Scioto River. With the large number and varied types of chemicals (some of which go undetected) discharged into Ohio rivers like the Ottawa, measures that integrate the effects of all important physical and chemical perturbations are needed to accurately and realistically examine trends in water resource quality. The case examples will also be more easily understood after the reader has become familiar with the concepts and terms from Ohio EPA's biocriteria framework described in Chapter 9.

2.1.1 Scioto River

The Scioto River mainstem downstream from Columbus has been monitored frequently since 1979 over a distance of approximately 40 miles. The purpose of this monitoring has been to document changes as the result of the upgrading of the two major WWTPs, Jackson Pike (RM 127.2) and Columbus Southerly (RM 118.4). A major combined sewer overflow (CSO) discharges 2.6 miles upstream from the Jackson Pike WWTP (RM 129.8). Columbus also withdraws water for drinking purposes at RM 134. This effectively leaves the mainstem between this intake (RM 134.0) and the Jackson Pike WWTP (RM 127.1) with very little flow during the summer and early fall months. Two low-head dams form sizable impoundments in this area as well. CSOs discharge mostly during wet weather (dry weather overflows

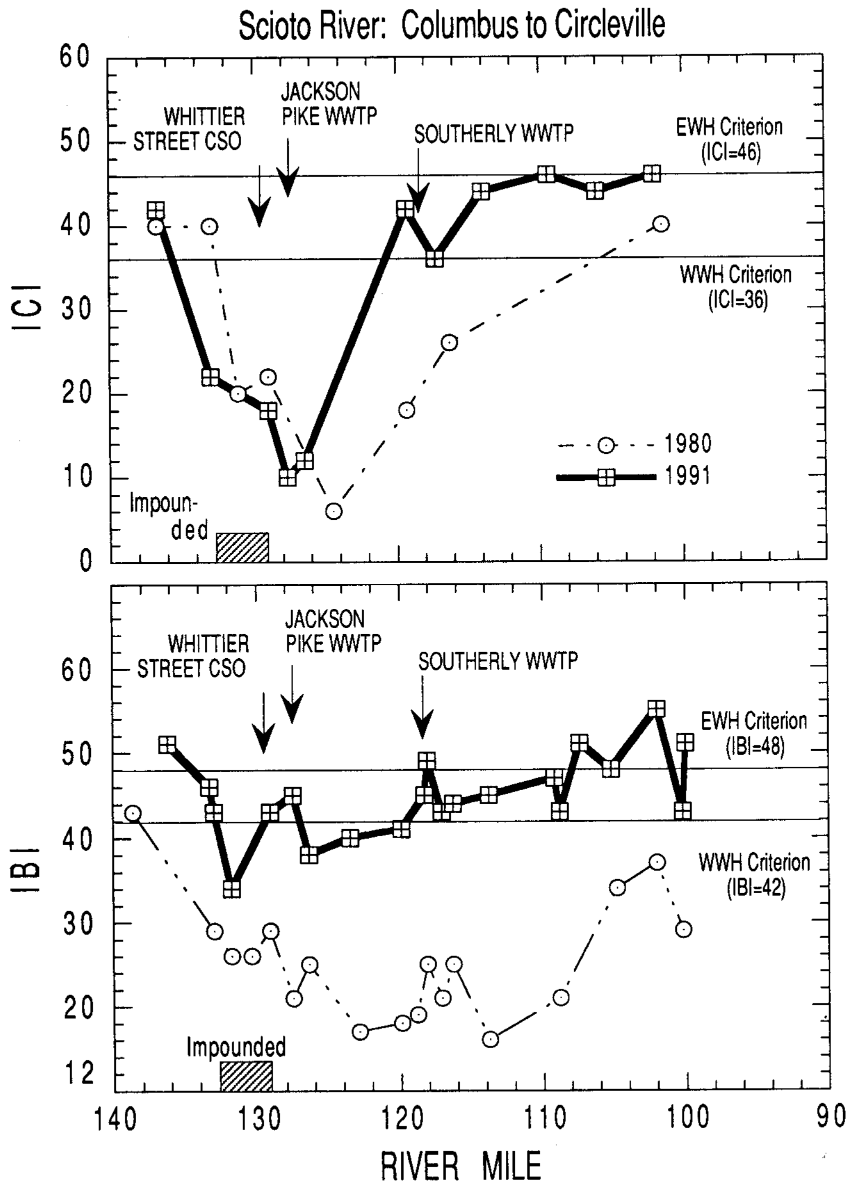


Figure 2. Longitudinal profile of the ICI (upper) and IBI (lower) in the Scioto River between Columbus and Circleville, Ohio during 1980 (circles) and 1991 (squares).

do exist, however) in the lower Olentangy River (confluence at RM 132.3) and to the mainstem between this point and the Jackson Pike WWTP. This arrangement of CSOs, impoundments, and water withdrawals is commonplace in Ohio cities and towns throughout the Eastern Corn Belt Plains, Huron/Erie Lake Plain, and portions of the Erie/Ontario Lake Plain ecoregions. Despite the aforementioned impoundments and flow alterations, overall habitat conditions in the Scioto River are good to excellent.

The Scioto River perhaps represents one of the best success stories of any river or stream in Ohio. Historically degraded since the early 1900s (Trautman 1981), a significant improvement in the fish community was evident between 1980 and 1991 (Figures 2 and 3). However, the macroinvertebrate community did not respond as quickly, although significant improvement was finally evident in 1991.

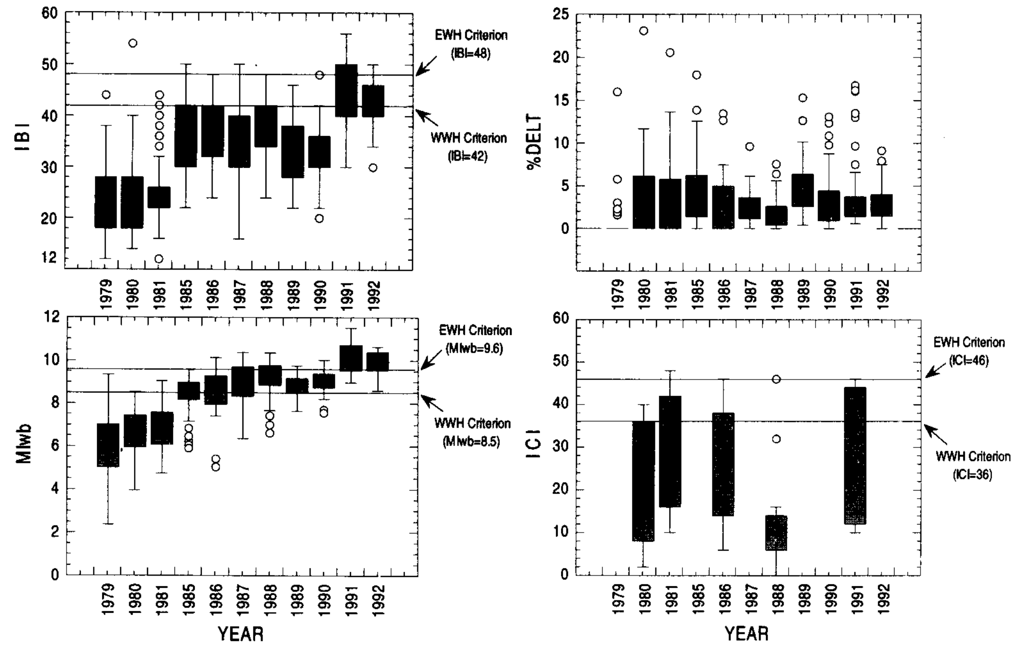


Figure 3. Boxplots of percent DELT anomalies on fish (upper left), MIwb (lower left), IBI (upper right), and ICI (lower right) for the Scioto River between Columbus and Circleville, Ohio for all years sampled between 1979 and 1992. Boxplots include the median (50th percentile), upper quartile (75th percentile), lower quartile (25th percentile), maximum, minimum, and outliers (values more than two interquartile ranges from the median).

Figure 3 portrays, in box-and-whisker plots by year, all 11 years of monitoring and reveals the broad improvement in the IBI and modified Index of Well-Being (MIwb) throughout the study area. The ICI trend has been less consistent and demonstrates the uncharacteristic lagging performance of this group in the segment most directly impacted by CSOs. Another important pattern with this example is the reversal in the response of the fish and macroinvertebrate communities (Figure 3). The years 1988 and 1991 were abnormally dry and both had extremely low flows. Conversely, rainfall and river flows during 1981 and 1986 were closer to normal. Although the ICI did not meet the Warmwater Habitat (WWH) biocriterion upstream from RM 109.4 in any year prior to 1991, the severity of the degradation was less in the normal flow years (Table 1). This was especially evident in the segment directly affected by the CSO discharges (Figure 2).

The combined effect of the upstream water withdrawals, impoundments, and remnants of past channel deepening have resulted in a physical environment that resembles a series of ponds during dry weather periods. The pools downstream from the Whittier Street CSO are unusually deep, which, combined with the artificially manipulated flow conditions, can result in an extremely long turnover rate. The duration of water turnover in each pool is correlated with the duration of the dry weather, low-flow periods. It is during these times that the products of the CSO discharges, urban runoff, and enrichment from upstream agricultural sources become concentrated in this area and the result is an extremely enriched aquatic environment. Attached algal growths completely smother and embed the coarse cobble/gravel substrates in the shallower run and riffle areas between pools and immediately downstream. The macroinvertebrate community in particular is composed of pollution tolerant taxa typical of this type of organic enrichment and, often, very high organism densities are observed. This is a key response signature of this type of impairment. The fish community has been less impacted than the macroinvertebrates as both the MIwb and IBI perform at least in the fair range and marginally met the WWH criteria in 1991.

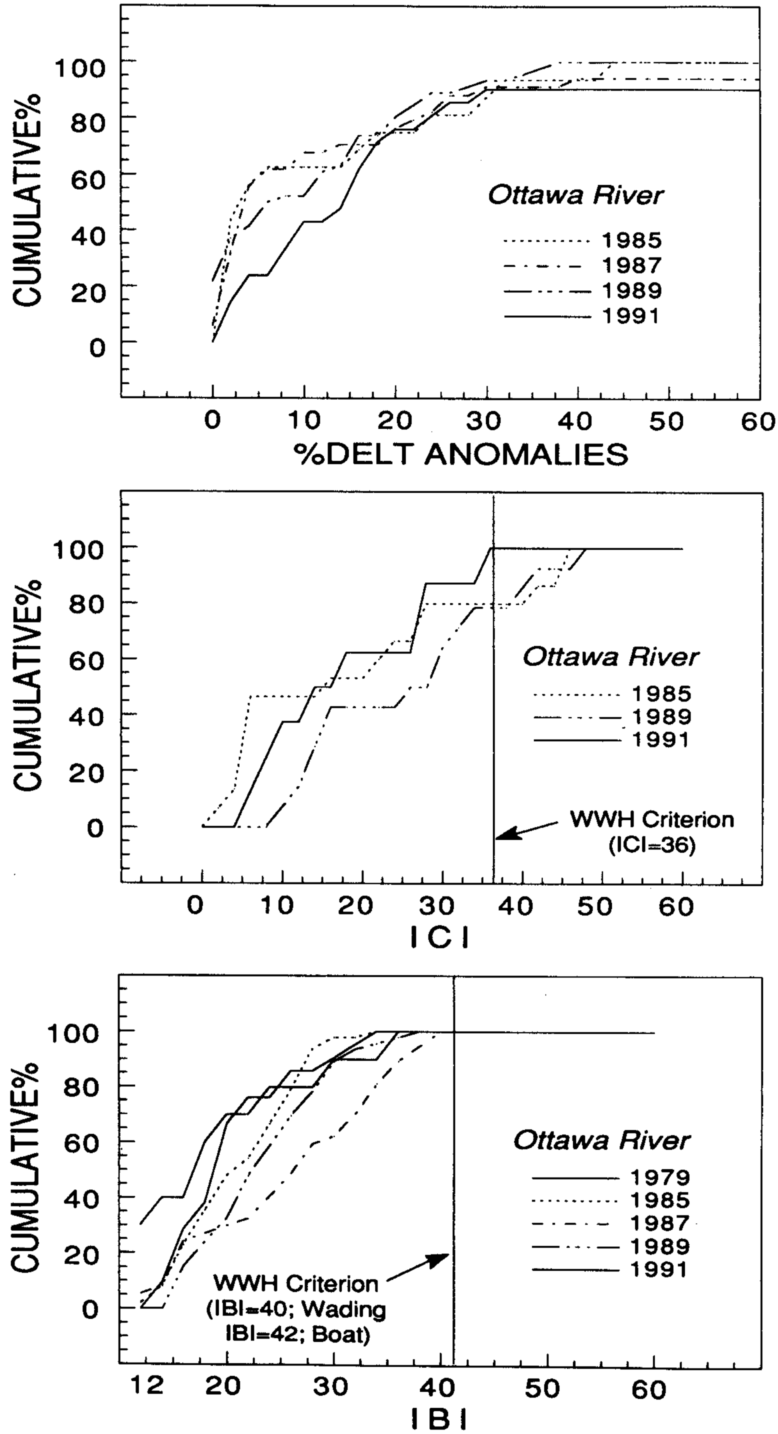


Figure 4. Cumulative frequency distribution of percent DELT anomalies in fish (upper), ICI (middle), and IBI (lower) in the Ottawa River between Lima, Ohio and the mouth based on data collected during five years between 1979 and 1991.

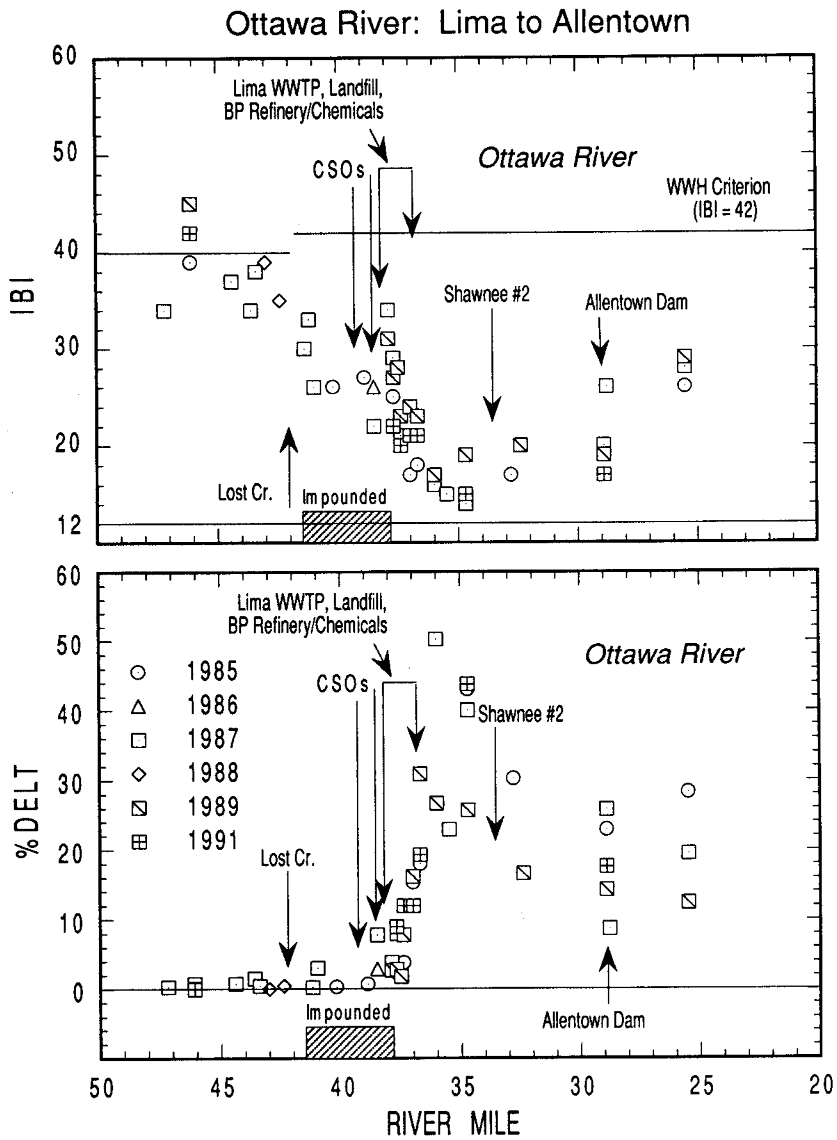


Figure 5. Longitudinal plot of all values for the IBI (upper) and percent anomalies on fish (lower) from Lima to Allentown, Ohio for six years between 1985 and 1991.

However, the incidence of external anomalies (mostly eroded fins and lesions) has been elevated (>5 to 13%) downstream from the CSOs and both WWTPs, and fish community indices (IBI and MIwb) tended to be lower during the high-flow years of 1989, 1990, and 1992.

The recovery exhibited by both organism groups since the late 1970s is best correlated with reduced loadings of sewage at both WWTPs and, to a lesser extent, the CSOs. The fish community showed a substantial response to the near elimination of raw sewage bypassing at the Columbus Southerly WWTP after 1982. The improvements observed in 1988 include partial attainment of the Exceptional Warmwater Habitat (EWH) criteria in the lower section of the study area. In 1991, full attainment of EWH was observed upstream as far as RM 109.4, and partial attainment at RM 114.0. This would likely increase if siltation impacts from nonpoint sources could be reduced. Historically, the Scioto River has been severely degraded with the entire mainstem from Columbus to the Ohio River having a dissolved oxygen

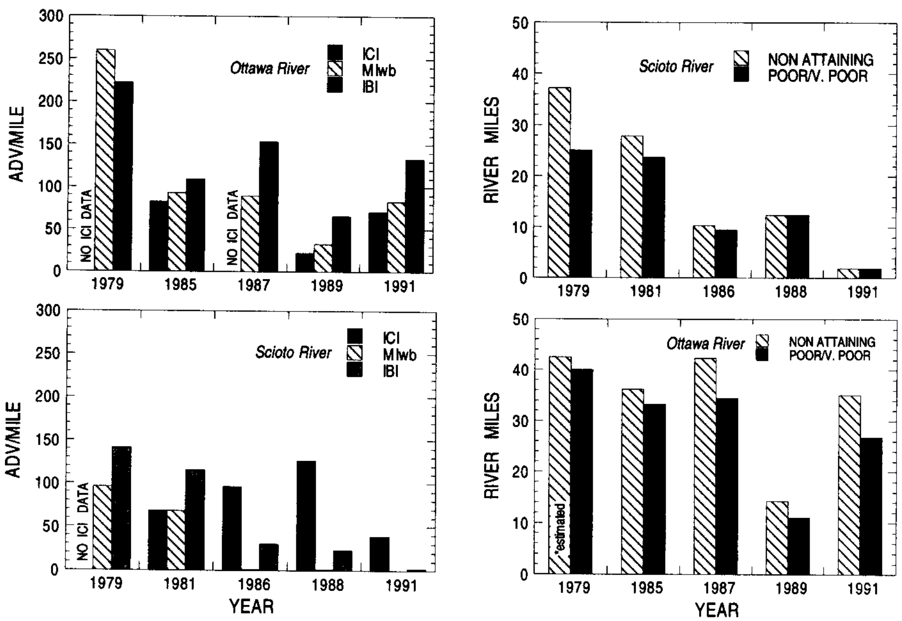


Figure 6. Comparison of ADV-per-mile statistics for the ICI, MIwb, and IBI from the Ottawa River (upper left) and Scioto River (lower left), and the miles of WWH nonattainment and miles of poor and very poor performance for the Ottawa River (upper right) and Scioto River (lower right).

content of near zero (and presumably very degraded aquatic communities) as recently as the late 1950s. In 1991, the total non-attaining miles shrank from 37.2 in 1979 to only 1.9 in 1991 (Table 1). The miles in full attainment of WWH increased markedly from 2.1 miles in 1979 to 24.1 miles in 1991, whereas the miles in partial attainment held steady after 1981. Thus, the increase in full attainment miles has resulted largely from a reduction in non- and partially attaining miles. While these observations demonstrate the beneficial effect of reduced loadings from the traditionally regulated point source discharges, underlying impacts are beginning to emerge. These include organic enrichment due to CSO discharges and urban runoff (which can be exacerbated by altered hydrology and habitat), and siltation from agricultural and urban sources.

2.1.2 Ottawa River

The Ottawa River in and downstream from Lima has been monitored frequently by Ohio EPA since the early 1970s, and infrequently since the turn of the century. The principal point sources include the Lima WWTP and CSOs, and two industrial facilities, a petroleum refinery, and a chemical manufacturing plant. There are similarities between this area and the Scioto River in that each river flows through an extensively urbanized area, each is impacted by CSOs, each has little upstream flow for dilution of point sources during dry weather, low-flow periods, and both are dominated by effluent flow downstream from the major point sources. The key difference is with the contributions from the two major industrial sources to the Ottawa River, which are essentially lacking in the Scioto River. Significant improvement in the biological indices has been observed during the past twenty years (Figure 4; Table 1), which has been due in large part to significant reductions in point source loadings. The improvement is even more remarkable when conditions since the turn of the century, when the river lacked any fish life for a distance of nearly 40 miles (Ohio EPA 1992a), are considered. However, the results of biological sampling conducted between 1985 and 1991 show that the Ottawa River has not attained full recovery in terms of the WWH biocriteria (Figure 4; Table 1). The most severely impaired segment remains downstream from the Lima WWTP and the British Petroleum (BP) Oil refinery and BP Chemicals facility where aquatic community performance remains poor. Compared to the Scioto River, the Ottawa River exhibits evidence that, despite some improvements, makes it one of the most severely impaired rivers in the state.

Table 1. Area of Degradation Value (ADV) Statistics for Selected Sampling Years in the Scioto River and Ottawa River Study Areas

Stream Index	Biological index scores: Upper RM	Lower RM	Minimum	Maximum	ADV statistics: ADV[2]	ADV/ Mile[3]	Poor/VP ADV[4]	Attainment status (miles[1]): Full	Partial	Non	Poor/VP[5]
Scioto River (1979)											
ICI (no data available)											
MIwb	140.0	100.2	4.0	8.7	3810	96	227	2.1	0.9	37.2	25.1
IBI			12	42	5618	141	1403				
Scioto River (1981)											
ICI			10	48	2691	68	151				
MIwb	140.0	100.2	5.2	8.4	2695	68	68	1.7	10.6	27.9	23.7
IBI			18	43	4582	115	541				
Scioto River (1986)											
ICI			6	46	3155	96	272				
MIwb	133.0	100.2	7.4	9.8	35	1.1	0	10.1	13.3	10.3	9.5
IBI			27	45	980	30	0				
Scioto River (1988)											
ICI			0	46	4147	126	689				
MIwb	133.0	100.0	7.4	10.0	35	1.1	0	9.7	11.6	12.4	12.4
IBI			28	47	750	23	0				
Scioto River (1991)											
ICI			10	46	1432	39	18				
MIwb	136.3	100.0	8.3	11.2	0	0	0	24.1	11.0	1.9	1.9
IBI			30	55	84	2.3	0				
Ottawa River (1979)											
ICI (no data available)											
MIwb	46.0	22.9	0.7	8.3	6015	260	806	0.6	0.0	23.6	22.2
IBI			12	36	5117	222	2255				
Ottawa River (1985)											
ICI			2	46	3734	82.4	532				
MIwb	46.1	0.8	2.3	8.8	4200	92.7	259	3.9	6.3	36.2	33.2
IBI			15	39	4959	109	1610				
Ottawa River (1987)											
ICI (no data available)											
MIwb	46.1	1.2	3.2	8.9	4025	89.6	296	3.1	0.6	42.3	34.4
IBI			14	42	6868	153	3757				
Ottawa River (1989)											
ICI			10	48	1001	22.1	7				
MIwb	46.1	0.8	3.9	9.0	1470	32.5	103	17.4	14.7	14.3	11.0
IBI			17	45	2978	65.7	517				
Ottawa River (1991)											
ICI			6	36	3209	70.8	85				
MIwb	46.1	0.8	3.4	9.2	3755	82.9	287	2.0	9.3	35.1	26.8
IBI			15	42	6019	133	1345				

Note: ADV is used here to demonstrate changes in the extent (miles) and severity (departure from the criterion for each index) of biological community impairment during representative survey years both before and after pollution abatement efforts at major point source discharges.

1 Includes extrapolation of the results for a limited distance upstream and downstream from the upper and lower extremes of the study area.
2 ADV: the Area of Degradation Value calculated for the study area.
3 The average ADV per mile, which normalizes comparison between different years and study areas of differing length.
4 The portion of the total ADV that is contributed by poor and very poor (VP) index values.
5 The nonattaining miles within which biological index values are in the poor or very poor (VP) range of performance.

The combination of low fish species and macroinvertebrate taxa richness, a predominance of toxic tolerant species and taxa, and an unusually high incidence of external anomalies on fish point to a residual toxic impact in concert with low dissolved oxygen and organic enrichment. The incidence of deformities, eroded fins, lesions, tumors (DELT) anomalies was highest immediately downstream from the three major point sources exhibiting a precipitous increase (Figure 5). This corresponded to an almost equally precipitous decline in the IBI and MIwb. The ICI also exhibited the lowest scores in this area. A cumulative frequency distribution (CFD) analysis of different data aggregations (Figure 4), shows that the highest IBI scores occurred in 1987 and 1989, the lowest percent DELT in the same years, and the highest ICI in 1989 (no ICI results in 1987). Thus, the results from the most recent year (1991) have actually declined unlike what has been observed in the majority of streams and rivers throughout the state. In addition to the permitted point source discharges, spills and unpermitted discharges have been a prominent factor with 31 **reported** incidents since 1985. In addition, the cumulative pattern of noncompliance with NPDES permit limits by the three major point sources between 1989 and 1991 shows that there is a high probability that one or more of the sources will record at least one violation in any given month. Although none of the three sources is individually considered to be in significant noncompliance, the cumulative frequency of violations would constitute a significant violation. While the chemical sampling results showed exceedences of chemical water quality criteria mostly for dissolved oxygen and ammonia-N (chronic thresholds only) this alone did not indicate the full magnitude of the biological problems. Sediment chemistry results showed highly to extremely elevated heavy metals in the nine-mile long segment with the highest biological stress indications and biochemical markers from individual fish showed stress responses to organic chemicals (Ohio EPA 1992a). Thus, the cumulative frequency of NPDES violations, spills and other episodic discharges, elevated sediment metals, and low dissolved oxygen combine to create frequently occurring episodes of stress, which are much more apparent in the biological results than the chemical water column results. These results also point to the difficulties in relying on NPDES permit compliance **alone**, especially in complex areas such as the Ottawa River.

2.2 Trend Assessment

The statistics generated by the ADV and the longitudinal analysis of the community indices for different years provides information to evaluate the magnitude of change over time using several different dimensions of the change. Not only can the relative comparisons of magnitude (i.e., ADV statistics) be made, but the extent of various levels of condition (miles of nonattainment and miles of poor and very poor performance) can also be portrayed (Table 1). This is graphically illustrated in Figure 6 where the ADV per mile, the miles of nonattainment of the applicable aquatic life use designation, and the miles of poor or very poor performance for the Scioto River and the Ottawa River for selected sampling years during 1979 through 1991 are compared. Although the rivers are of different sizes, the relative magnitude of the impacts from point sources, flow diversions, and urbanization are similar. From the information portrayed in Figure 6 the Ottawa River is and has been more severely impacted than the Scioto River as indicated by higher ADV per mile values for each of the three indices. Also, whereas the decline in the ADV per mile has been consistent for the Scioto River, it has been much less so for the Ottawa River. This is even more evident in the comparison of the nonattaining and poor/very poor miles where the number of these miles was nearly eliminated in the Scioto River by 1991. The Ottawa River showed improvement in this category through 1989, but both categories increased in 1991. This information, combined with the biological community responses and knowledge of the sources and environmental setting, shows that the Ottawa River continues to be impaired by toxic impacts. The Scioto River has never had any significant problem with toxics, which is certainly evident in the biological responses. Problems do remain, but all of the indicators point to conventional problems associated with the municipal sewer system, particularly CSOs.

The preceding examples show how biosurvey information and biocriteria function as a direct measure of biological integrity, provide an unambiguous goal for a waterbody, and provides a way for water quality managers to visualize ambient conditions and evaluate the effectiveness of administrative programs and decisions. It also allows a quantification of observed departures from clean water goals beyond the traditional pass/fail framework. Integration of the trend assessment and ADV-based approaches outlined above into existing water quality management programs should increase accuracy, sensitivity, and discrimination in assessments of water resource integrity. This, in turn, should result in a more defensible allocation of funds and more efficient use of resources for accomplishing realized water quality improvements.

3.0 ABILITIES AND LIMITATIONS OF BIOSURVEY RESULTS TO DISCRIMINATE IMPACTS: BIOLOGICAL RESPONSE SIGNATURES

The availability of a comprehensive, standardized ambient biological database from a variety of environmental settings in Ohio has permitted certain patterns and characteristics of biological community response to perturbations to be identified. A common criticism of ambient biological survey data has been of its inability to determine the cause or source of an impaired condition (USEPA 1985; Suter 1993). While this is probably a valid concern for single-dimension indices (e.g., $\bar{d}$, $\overline{H}$, number of species, biomass, etc.) it does not apply equally to the new-generation multimetric indices such as the Index of Biotic Integrity (IBI) and Invertebrate Community Index (ICI).

When the response patterns of the various metrics and components of these indices were examined from areas where the predominant impairment causes and sources are well known, some consistent patterns emerged. Unique combinations of biological community characteristics that aid in distinguishing one impact type over another are referred to as *Biological Response Signatures.* These proved valuable in delineating predominant causes and sources of the aquatic life use impairments identified in the 1990 and 1992 305(b) reports (Ohio EPA 1990b, 1992b) and the many basin/subbasin assessments accomplished each year.

The original effort included a database made up of 25 similarly sized streams and rivers (drainage area range 90 to 450 sq. mi.) from the Eastern Corn Belt Plains (ECBP) and Huron/Erie Lake Plain (HELP) ecoregions (Yoder 1991). The data were from surveys conducted between 1982 and 1989 and which followed Ohio EPA procedures (Ohio EPA 1987a,b, 1989a,b). One of six general impact types were assigned to each sampling site. A parallel effort evaluated techniques by which even more subtle differences might be further defined within the concept of Biological Response Signatures (Anderson et al. 1990). This involved the use of genetic algorithms employing artificial intelligence and machine learning techniques. One initial finding of this effort was the distinctive response of the sensitive species metric. This metric of the headwaters IBI combines the intolerant metric of the wading and boat site type IBIs with moderately intolerant species (Ohio EPA 1987b). This aggregation of the community data was by itself found to consistently indicate the Complex Toxic impact type with a reliability of 82% in the stream and river sizes outside of its assigned use in the headwaters IBI.

This original analysis is updated here to include a database of more than 1200 samples from more than 70 streams and rivers spanning the period from 1981 through 1992. Although 20 different impact types have been classified thus far, eight of the impact types were assessed in this analysis. These represent the most common types of impacts that occur in Ohio rivers and streams. In addition, this analysis was limited largely to the Eastern Corn Belt Plains and Huron/Erie Lake Plain ecoregions in order to minimize the influence of this important factor. The nine impact types used in the analysis are described as follows:

1. **Complex Toxic** — Impacts from the complex combination and interactions of major municipal WWTP and industrial point sources that comprise a significant fraction of the summer base flow of the receiving stream *and* where one or more of the following have occurred: serious instream chemical water quality impairments involving toxics, recurrent whole effluent toxicity, fish kills, or severe sediment contamination involving toxics has occurred. This may include areas that have combined sewer overflows (CSOs) and/or urban areas located upstream from the point sources.
2. **Conventional Municipal/Industrial** — Impacts from municipal WWTPs that predominantly discharge conventional substances (these may or may not dominate stream flows) **and** where no serious or recurrent whole effluent toxicity is evident **or** small industrial discharges that may be toxic, but which do not comprise a significant fraction of the summer base flow; other influences such as CSOs and urban runoff may be present upstream from the point sources.
3. **Combined Sewer Overflows/Urban** — Impacts from CSOs and urban runoff within cities and metropolitan areas that are in direct proximity to sampling sites. This includes both free-flowing and impounded areas **upstream** from the major WWTP discharges. Minor point sources may also be present in some areas.
4. **Channelization** — Areas impacted by extensive, large-scale channel modification projects and where little or no habitat recovery has taken place. Some minor point source influences may be present.
5. **Agricultural Nonpoint** — Areas that are principally impacted by runoff from row-crop agriculture, which is the predominant agricultural land use in the ECBP and HELP ecoregions. Some minor point source and localized habitat influences may be present.
6. **Flow Alteration** — Sites affected by flow alterations including controlled flow releases downstream from major reservoirs or areas affected by water withdrawals as the predominant impact.
7. **Impoundment** — River segments that have been artificially impounded by low-head dams or by flood control and water supply reservoirs and where this is the predominant impact type present.

8. **Combined Sewer Overflows/Urban with Toxics** — The same as impact type 3 (CSO/Urban Conventional) except that there is a significant presence of toxics. Included are municipal CSO systems with significant pretreatment programs and sources of industrial contributions to the sewer system.
9. **Livestock Access** — Sites directly impacted by livestock operations where the animals (mostly cows) have unrestricted access to the adjacent stream.

These impact types were assigned based on our knowledge of the types of pollution and disturbance sources and other chemical (water column, sediments), physical (habitat), and toxicological (bioassays) information from Ohio EPA's various water and effluent databases. This information is contained in basin/subbasin specific reports, the biennial water resource inventory (305b report), and the Ohio EPA Water Body System. One of these was assigned as the primary impact type to each of the more than 1200 sites sampled for fish and macroinvertebrates. Assignments were based on the predominant impact that was directly influencing the site at the time the sampling took place. The assignments were also based on the site-specific knowledge of the study area gained by Ohio EPA while conducting biological surveys of the 70-plus streams and rivers. The extent of spatial overlap between different impact types throughout the database is somewhat variable ranging from a clear predominance of a single impact type to the overlapping influence of two or three impact types. Secondary and ancillary impact types were simultaneously assigned in these instances. The key objective of this analysis is to determine whether or not the feedback gained from the biological community can communicate about and characterize these differences. The results are provided in Figures 7 through 12. This project, thus far, has concentrated on two- and three-dimensional analyses of IBI, MIwb, and ICI metrics and selected subcomponents.

3.1 Fish Community Responses

Three components of the fish community data were included in three-dimensional plots that illustrate the concept behind examining the combined biological response characteristics of each impact type. The IBI, MIwb, and the frequency (%) of DELT anomalies on individual fish are different expressions of the relative health of the fish community at a given location. The Complex Toxic impact type was compared on a three-dimensional basis to each of 7 other impact types (Figure 7). In each comparison the Complex Toxic impact type exhibited a fairly distinct clustering pattern as compared to the other impact types. The amount of overlap was least with the Flow Regulation and Agricultural Nonpoint Source types and greatest with the CSO Toxic impact type. The response characteristics that most typified the Complex Toxic impact type includes IBI <18 to 22 (median and 75th percentile values), MIwb <4.5 to 5.9, and DELT anomalies >15.1% (median value), the combined occurrence of which should uniquely characterize a site impacted by sources which were characterized as Complex Toxic. The CSO Toxic is the only impact type to have IBI and MIwb values that exhibited any substantial overlap with the Complex Toxic. The key distinguishing characteristic was that the CSO Toxic impact type had lower percent DELT values (Figures 7 and 8). This overlap is not surprising given that many of the CSO Toxic impacted segments were in the same streams and rivers as some of the Complex Toxic impacted segments. Toxic substances discharged by industrial and other sources to the local WWTP also can enter the receiving streams and rivers via the CSOs. The other impact types also had some metric and biological index values that overlapped with the Complex Toxic impact type, but this was usually at the extreme ends of the ranges for each impact type and generally represented the transition from the Complex Toxic impact to another type of impact as distance from the source increased. Many of the Complex Toxic impacted sites with metric and index values that overlapped with other impact types were generally in the reaches of the various study areas where recovery was taking place along the longitudinal continuum. Thus, the differences between the Complex Toxic impact type and some of the other impact types are transitional, being more so for some types than others. In addition, even though one or more metrics or indices had overlapping values between the Complex Toxic impact and other types, very seldom do they have **all** characteristics in common. In the three-dimensional analysis (Figure 7) it was the percent DELT IBI metric that seemed to be the key discriminatory attribute in separating the Complex Toxic impact from the other impact types.

The classification of sites according to the previously described impact types was done strictly on a cultural basis, i.e., the selections were made based on the types of sources present in a study area. A few

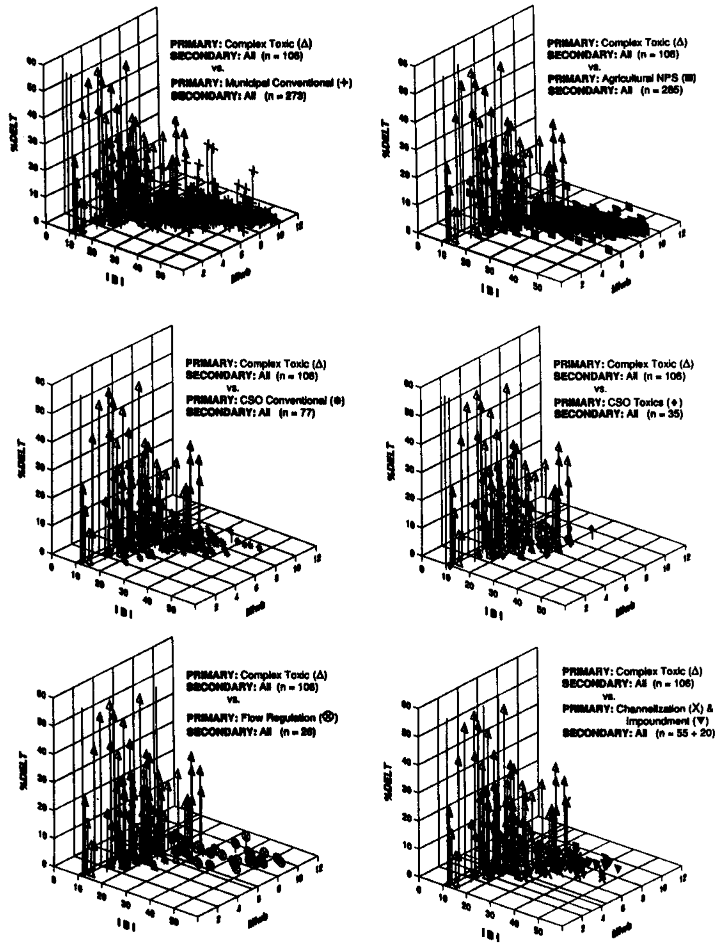


Figure 7. Three-dimensional comparison of the IBI, MIwb, and percent DELT anomalies among seven different impact types for 1214 electrofishing samples primarily from streams and small rivers in the ECBP and HELP ecoregions. The Complex Toxic impact type is compared to six other impact types described in the text. The sample size (n) is for primary and secondary impact types combined.

sites classified as the Agricultural Nonpoint Source impact type exhibited metric and index value combinations more consistent with the Complex Toxic impact type. Upon further investigation it was learned that one site was downstream from an experimental agricultural conservation tillage demonstration plot where pesticide usage was atypically intensive. The resultant biological response confirmed that the community response at this site better fit the Complex Toxic impact type in terms of the community response. Thus, the instream biota provided a more reliable indication of the type of impact than did the cultural characterization. Thus, the availability of the biological response signatures should fulfill the role as a guide for discovering and characterizing previously unknown problems.

We also examined all nine of the impact types in a two-dimensional framework. Some highlights of this analysis (Figures 8 and 9) of the IBI metrics and fish community indices are

1. Metric values and fish community index scores for the Complex Toxic and CSO Toxic impact types consistently indicated the lowest quality for the IBI, MIwb, darter species, percent round-bodied suckers, sensitive species, percent DELT anomalies, intolerant species, percent tolerant species, and density (less tolerant species).
2. Channelization had similarly low or even lower metric values for percent round-bodied suckers, intolerant species, sunfish species (lowest), percent top carnivores, percent simple lithophils, and

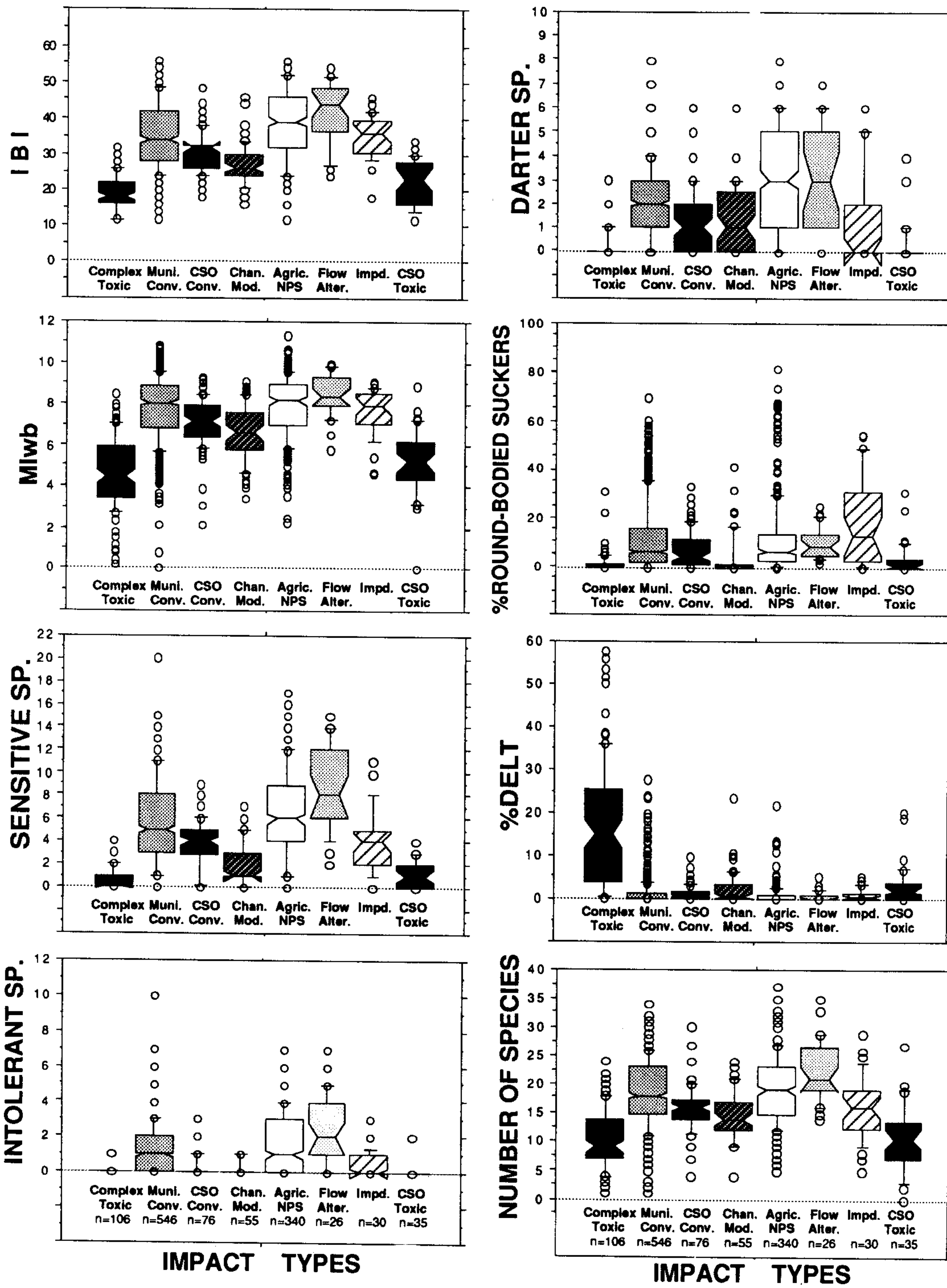


Figure 8. Notched box-and-whisker plots of six different IBI metrics, tne IBI, and MIwb by eight impact types for 1214 electrofishing samples primarily from streams and small rivers in the ECBP and HELP ecoregions.

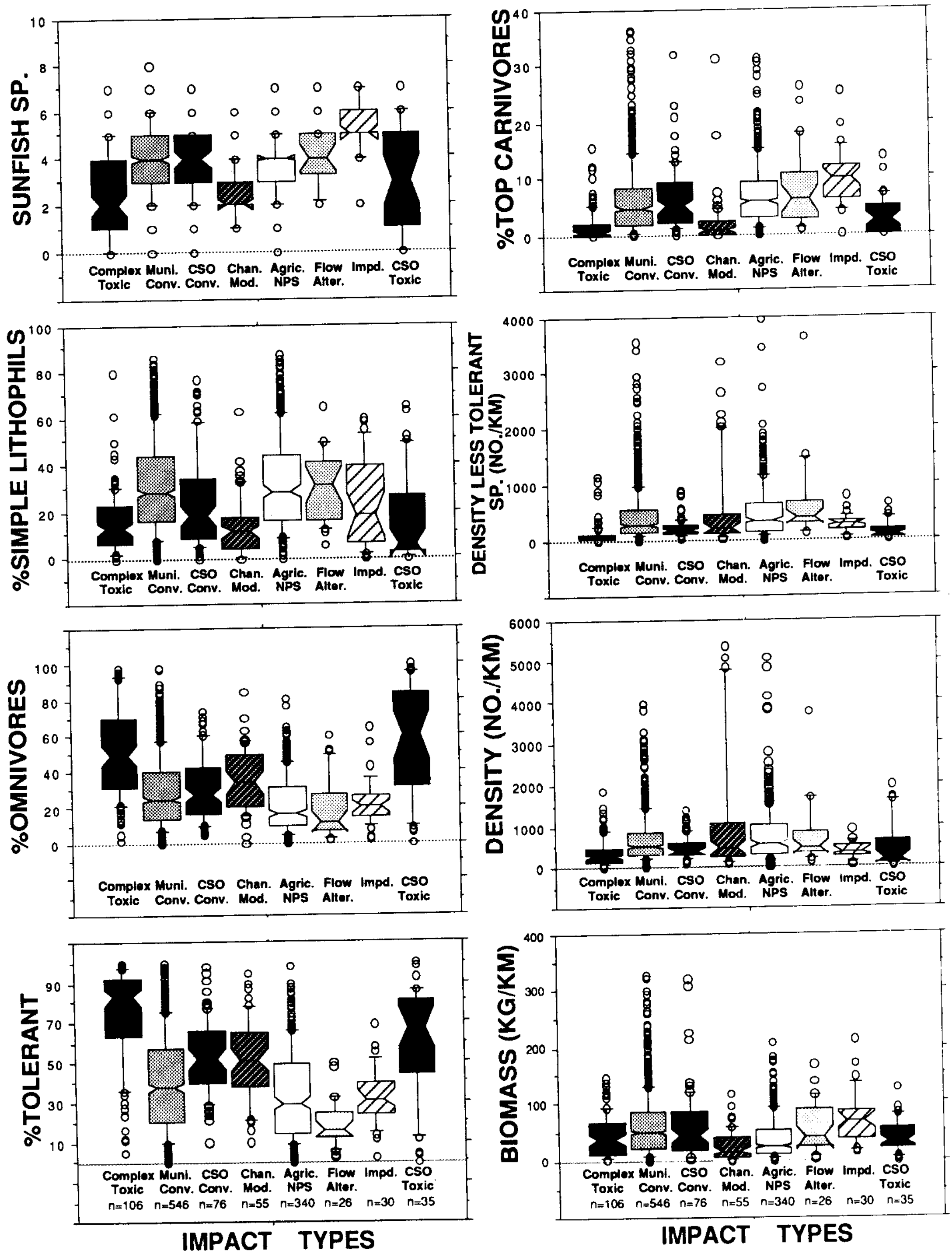


Figure 9. Notched box-and-whisker plots of six different IBI metrics, density (relative numbers), and biomass by eight impact types for 1214 electrofishing samples primarily from streams and small rivers in the ECBP and HELP ecoregions.

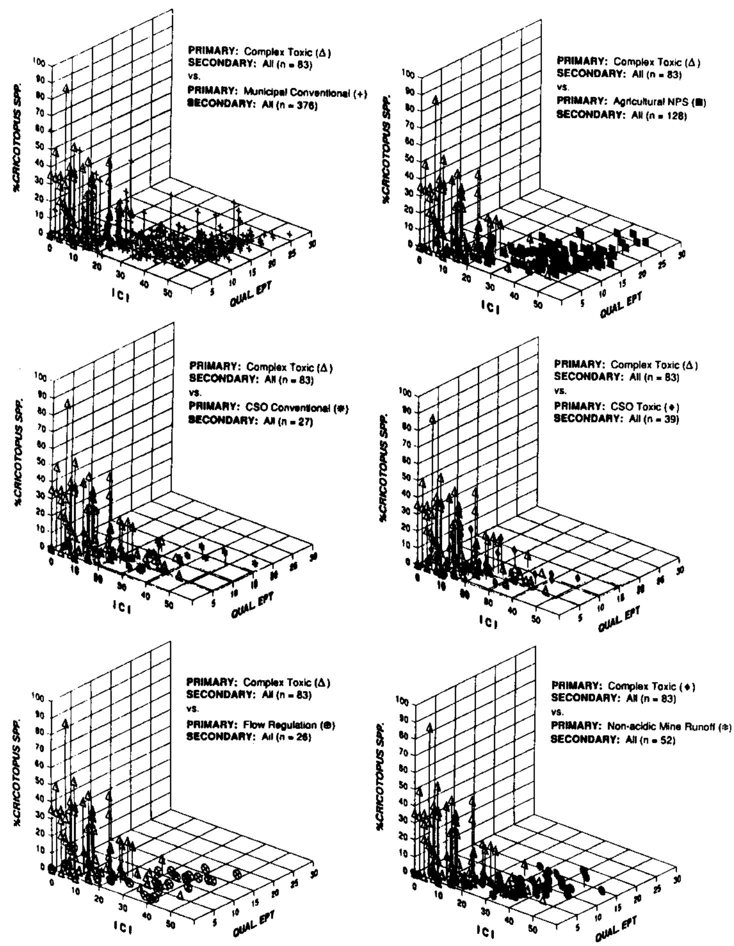


Figure 10. Three-dimensional comparison of the ICI, qualitative EPT taxa, and percent *Cricotopus* spp.among seven different impact types for 520 artificial substrate sampling sites primarily from streams and small rivers in the ECBP and HELP ecoregions. The Complex Toxic impact type is compared to six of the other impact types described in the text. The sample size (n) is for primary and secondary impact types combined.

biomass. In addition, Channelization exhibited some of the highest maximums and outliers for density (including tolerants). However, other metric values indicative of any toxic impacts were not evident for this impact type (e.g., percent DELT anomalies were very low).

3. Metric and index values indicative of good and exceptional performance were most frequently observed for the Flow Alteration and Agricultural NPS impact types followed by Conventional Municipal. Fair and poor performance was also observed under these types especially under extended low flows due to water withdrawals, higher effluent loadings, and/or more intensive land use and riparian impacts.
4. The incidence of extreme outliers was the most evident for the Municipal Conventional and Agricultural NPS impact types for darter species, percent round-bodied suckers, percent DELT anomalies, intolerant species, number of species, percent top carnivores, percent simple lithophils, density (less tolerants), density (including tolerants), and biomass. This indicates a wide range of biological performance within these two impact types from exceptional to poor due to the variable character of the local and segment-specific impacts. The extreme range of outliers in the Municipal Conventional impact type for percent DELT anomalies greater than 10% was observed mostly within or in close proximity to WWTP mixing zones.

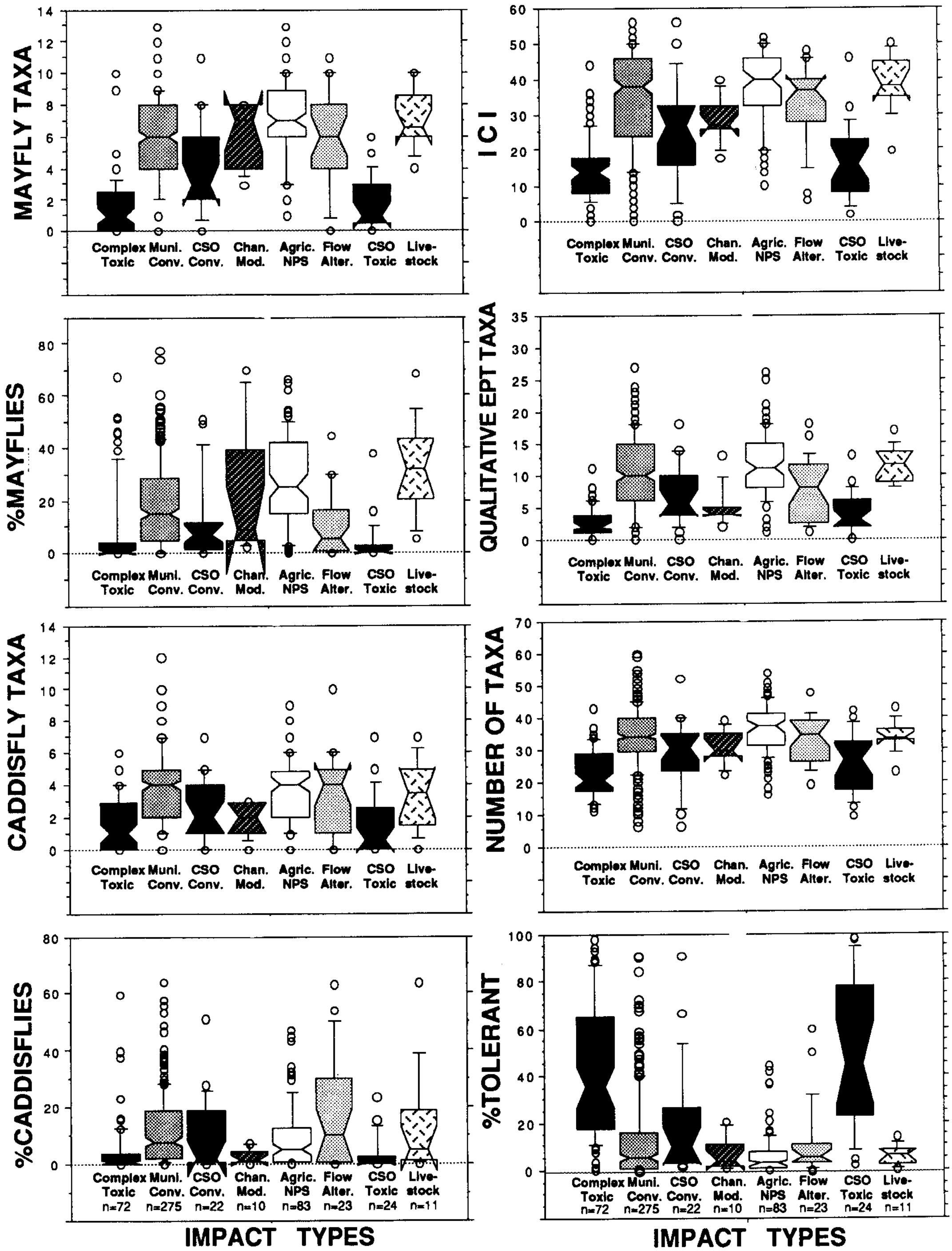


Figure 11. Notched box-and-whisker plots of seven different ICI metrics and the ICI by eight impact types for 520 artificial substrate sampling sites primarily from streams and small rivers in the ECBP and HELP ecoregions.

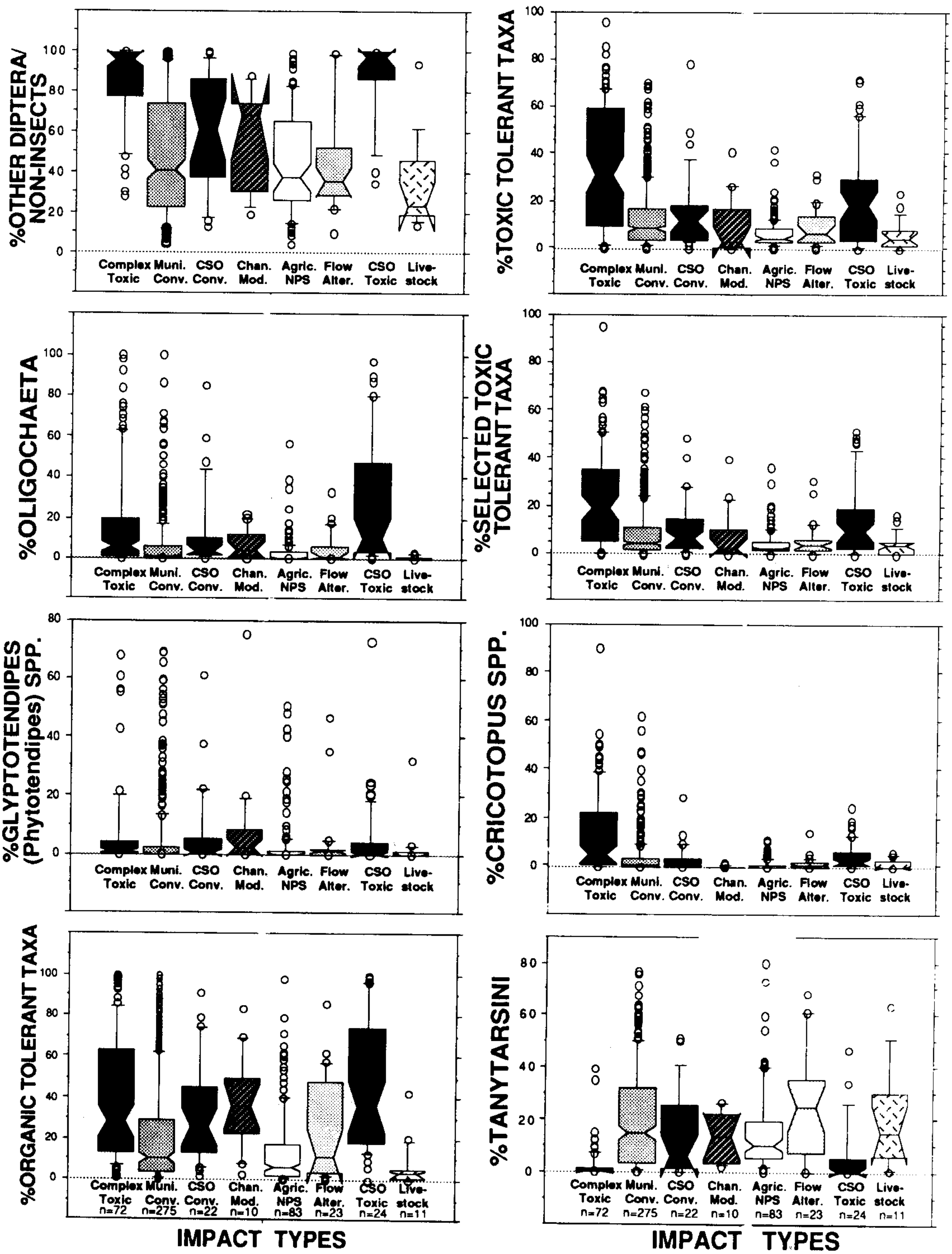


Figure 12. Notched box-and-whisker plots of two ICI metrics, percent toxic tolerant taxa (see text), percent selected toxic tolerant taxa (see text), percent *Glyptotendipes* (*Phytotendipes*) spp., percent organic tolerant taxa (see text), and percent *Cricotopus* spp. by eight impact types for 520 artificial substrate sampling sites primarily from streams and small rivers in the ECBP and HELP ecoregions.

3.2 Macroinvertebrate Community Responses

For macroinvertebrates we examined the relationship between the ICI, qualitative EPT taxa, and several other ICI metrics and other subcomponents on a three-dimensional basis for seven of the impact types. These included two of the ICI metrics (percent tolerant taxa and percent Tanytarsini midges) and three other macroinvertebrate community aggregations including percent Oligochaetes, percent toxic tolerant taxa*, and percent *Cricotopus* spp. Of the macroinvertebrate community aggregations the percent *Cricotopus* spp. seemed to differentiate the Complex Toxic impact type better than any other ICI metric or aggregation tested, much like the percent DELT anomalies did for the fish community results. The three-dimensional plots revealed a fairly reasonable separation of the Complex Toxic impact type from the six other impact types tested (Figure 10). For the Complex Toxic impact type the response characteristics include the combination of an ICI <14 to 18 (median and 75th percentile values), Qualitative EPT <2 to 4 (median and 75th percentile values), and percent *Cricotopus* spp. >5% (median value), which should uniquely characterize a site impacted by sources that were characterized as Complex Toxic. As with the fish community, the closest overlapping impact type was CSO/Toxic, with this impact type having a lower percent *Cricotopus* spp. Members of this latter group were found to comprise a toxic assemblage indicative of toxicity in North Carolina streams (Eagleson et al. 1990). However, Barbour et al. (1991) point out that *Cricotopus* (*Nostocladius*) spp. is an important component of pristine, western montane streams and is atypical of other members of this genus with regard to pollution tolerance. Simply eliminating this taxon would sufficiently protect the analysis where it is present.

This analysis demonstrates the need to access and interpret community information beyond the ICI and the individual metrics. Yoder (1991), using a smaller data set, showed the relationship between the ICI and organism density (number per sq. ft.) in a two-dimensional framework to demonstrate this point. In the comparison of the Complex Toxic and CSO/Urban impact types the ICI alone yielded similarly low results for each. Thus, the ICI alone was unable to discriminate between these impacts. However, organism density, which is not a direct component of the ICI, yielded a separation of the two impact types.

We also examined all nine of the impact types in a two-dimensional framework. This included the ICI, nine of the ten ICI metrics, and six additional aggregations of the macroinvertebrate data (Figures 11 and 12). The analysis revealed the following:

- The ICI, qualitative EPT taxa, percent tolerant, mayfly taxa, percent mayflies, percent caddisflies, percent other dipteran/noninsects, percent Tanytarsini midges, percent *Cricotopus spp.*, percent selected toxic tolerant taxa, percent toxic tolerant taxa, and percent Oligochaeta consistently indicated the lowest quality for the Complex Toxic and CSO/Toxic impact types. The inclusion of Oligochaeta as a toxic tolerant group contrasts with the original findings of Brinkhurst (1965), but agrees with the findings of Eagleson et al. (1990) and others (Chapman et al. 1980; LaPoint et al. 1984).
- The ICI metrics percent caddisflies and qualitative EPT taxa were as low for the Channelization impact type as for the Complex Toxic and CSO/Toxic types. However, for many of the other ICI metrics and other aggregations the results were much different. This shows the need to examine the response patterns of a number of metrics and other aggregations of the community data.
- The percent *Glyptotendipes* (*Phytotendipes*) spp. showed a tendency toward higher values for the Conventional Municipal impact type, particularly as outliers, which would indicate a high degree of organic enrichment. This group also exhibited high values for CSO/Urban, Channelization, and in the form of outliers, the Agricultural NPS impact type. None of these appeared to be entirely distinctive and overlapped with the Complex Toxic and CSO/Urban impact types. The other aggregations used to indicate organic enrichment (percent Oligochaeta and percent organic tolerant taxa*) did not discriminate as well and were more indicative of the Complex Toxic and CSO/Toxic impact types.

* Two aggregations of toxic tolerant taxa were used: (1) percent toxic tolerant taxa includes all *Cricotopus* spp., *Dicrotendipes simpsoni*, *Glyptotendipes barbipes*, *Polypedilum* (P) *fallax* group, *Polypedilum* (P) *illinoense*, *Nanocladius distinctus*, *Hesomyia senata* or *Thienemannimyia norena*, Conchepelopia, *Natarsia* sp. A, and *Ferrissia* sp. and (2) percent selected toxic tolerant taxa includes *Cricotopus* spp., *Dicrotendipes simpsoni*, *Glyptotendipes barbipes*, *Polypedilum* (P) *fallax* group, *Polypedilum* (P) *illinoense*, and *Nanocladius distinctus*.

3.3 Implications for Minimum Dataset Requirements

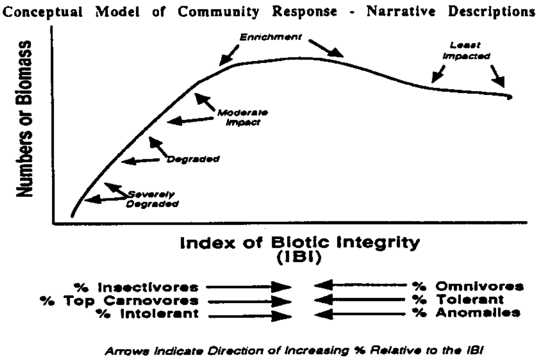
Minimum data requirements for using the Biological Response Signatures includes having sufficient information to employ the use of multimetric evaluation mechanisms, a standardized approach to data collection, and consistent and responsive management of the database. Other factors that further increase the analytical power of the biological data includes the use of multiple organism groups, an integrated approach to conducting assessments, and the inclusion of ancillary data such as biomass for fish. It is important to make these and other data collection decisions early in the process. An example of the importance of these early decisions was the recording of external anomalies on fish, which several years later allowed the development of the percent DELT metric of the IBI. This turned out to be a key metric in being able to discern the Complex Toxic impact type. At the time we did not realize that this use would exist; however, our failure to include it as a quantitative measurement early in the process would have constituted an unfortunate and irreplaceable loss of critical information. Another key decision was to identify midges to the genus/species level rather than limiting this to the family level. A failure to do this early on would have negated the use of the genus *Cricotopus* in the Biological Response Signatures. Thus, the ability to utilize biological data for diagnosis in Ohio was strengthened as the result of decisions made more than ten years ago not only about which organism groups to sample, but the types of information that were recorded. Frequently, biological monitoring programs are pressured to sacrifice data quantity and quality to meet regulatory and perceived financial constraints. As seen here, decisions made early in the process can have some far-reaching consequences in the long term.

4.0 CONCEPTUAL MODEL OF COMMUNITY RESPONSE

Another way to describe the attributes of ambient biological data for characterizing different types of environmental impacts is with a conceptual model. Figure 13 shows a model of the response of a fish community to increasing stress from a least impacted to severely degraded condition. The comparison of numbers and/or biomass with the IBI is used to demonstrate this conceptual relationship. The same general relationships would also apply to key macroinvertebrate community characteristics and the ICI. Beneath the graphic are narrative descriptions of biological community characteristics, chemical conditions, physical conditions, and examples of environmental perturbations that are typical of biological community responses across the five narrative performance classes. These are necessarily general and are not invariable. However, this model was developed from Ohio EPA's experience in analyzing biological, chemical, and physical data over a 15-year period and on a statewide basis and was further corroborated by the preceding analyses associated with the impact types and Biological Response Signatures. Thus the model has a good foundation in the observation of actual environmental conditions and associated biological community responses.

Results of the IBI from four similarly sized streams and rivers, each with different impact types and varying biological community performance, were plotted together (Figure 14) in an attempt further visualize these concepts. The results demonstrate the utility of using the theoretical range of the IBI (or ICI, MIwb, etc.) to differentiate between and interpret different types of impacts. The left column along the y axis in Figure 14 lists a gradient of impact types associated with the vertical scale of the IBI and the actual impacts present in each of the four areas listed in the right column. Lotic biological communities may experience spatially different impacts on a longitudinal basis with the degree of departure from the applicable biocriterion and recovery dependent on the severity and type(s) of impacts present; the Hocking River is one such example (Figure 14). Relatively unimpacted systems (e.g., Big Darby Creek) or those with moderate departures (e.g., Alum Creek) frequently reveal impairment within the fair range of performance while others may be uniformly devastated (e.g., Rush Creek), and reflect poor or very poor performance. These examples correspond to the narrative descriptions of community response and the attributes of the various impact types listed in the left column of Figure 14, and the range of

* Organic tolerant taxa includes the following: Oligochaeta, *Glyptotendipes* (*Phytotendipes*) spp., *Chironomus decorus* group, *Chironomus riparius* group, Turbellaria, *Physella* sp., *Simulium* sp., *Dicrotendipes lucifer*, *Dicrotendipes neomodestus*, and *Polypedilum* (Tripoda) *scalaenum* group.)



Degraded (Very Poor)	Severely Degraded (Poor)	Impact (Fair)	Moderate Enrichment (Good)	Least Impacted (Exceptional)
Community Conditions Characteristics				
No community organization - few or no species very low numbers only most tolerant, high % anomalies	Poorly organized community - few species, low numbers, tolerant species only, many anomalies	Reorganized community - tolerant species, intolerants in v. low numbers, omnivores predominate	Good community organization - good numbers of sensitive species, some intolerants	Highly organized community - insectivores, top carnivores, intolerants predominate, high diversity
Chemical Conditions				
Acutely toxic chemical conditions	Low D.O. with chronic toxicity	Low D.O., nutrient enriched, no recurrent toxicity	Adequate D.O., no acute/chronic effects, elevated nutrients	No effects evident, background conditions, good D.O.
and/or	*and/or*			
Habitat Conditions				
Total habitat loss, extremely contaminated sediments	Severe habitat degradation, severe sediment contamination	Modified stream channel, heavy siltation, canopy removal	Good habitat, no significant channel modifications	Excellent habitat, no modifications evident
Examples of Perturbations				
Toxic discharges, desiccation, acid mine drainage, severe thermal conditions	Municipal and industrial discharges, intermittent acute impacts	Municipal sewage, combined sewers, heavy agricultural use, non-acid mine drainage,	Minor sewage inputs, most agricultural non-point affected areas moderate thermal increase	No perturbations evident

Figure 13. Conceptual model of the response of the fish community to a gradient of impacts in warmwater rivers and streams throughout Ohio.

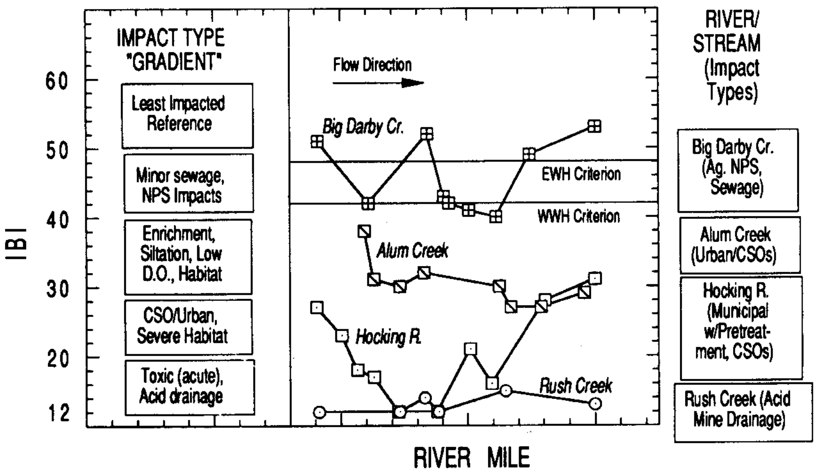


Figure 14. Biological response portrayed by the longitudinal trend in the IBI in four similarly sized Ohio rivers and streams with different types of point and nonpoint source impacts. Conceptual models of the impacts are also included on the left side under the heading Impact Type "Gradient."

performance described by the conceptual model (Figure 13). The important point here is that the biota integrates differing types and degrees of environmental impact along spatial and temporal scales and provides feedback that is inherently more accurate than we can predict or describe using cultural, surrogate, or process characterizations alone. Also, insight is gained on what can be expected as the predominant impact(s) in a particular segment change over time and/or space as a result of decreasing or increasing pollution levels (e.g., Hocking River). For example, we can expect a predominantly toxic impact to change temporally and spatially to a conventional impact when the sources of toxicity are controlled or eliminated. Different impact types are frequently "layered" in rivers and streams with the less severe impact types being masked by those that presently result in a more severe degradation (see Figure 1). As the more severe problems are reduced or eliminated the lesser problems may become evident in the results. Examples of this are presently being observed in Ohio streams and rivers where the abatement of historically severe municipal and industrial point source problems is leading to the unmasking of nonpoint source and other impacts (e.g., CSOs) that were always present, but not directly evident from the biological results.

Definite patterns in biological community data exist and can be used in the determination of whether or not a waterbody is attaining its designated use and in identifying the predominant associated causes of impairment. We have used this approach in producing the biennial 305b report, specifically the assignment of associated causes and sources of aquatic life use impairment (Ohio EPA 1990b, 1992b). While this does not represent a strict cause-and-effect analysis, it is intended to demonstrate causal associations based on a compilation of "lines of evidence." This involves utilizing all chemical, physical, biological, and source information as stressor, exposure, and response indicators to establish the most probable causes of observed biological impairments. Other uses of this approach include supporting enforcement and litigation proceedings and damage assessments. This information has been used to evaluate and verify claims made by NPDES permit holders that the degradation measured was due to factors (e.g., poor habitat or other sources) unrelated to their discharge. The Biological Response Signatures can be particularly useful in demonstrating that the observed degradation is likely related to specific discharges, especially those involving the Complex Toxic impact type. While the legal requirements of the Clean Water Act may be viewed as being sufficient to require entities to reduce pollutant loadings, the system of challenging these mandates requires the regulatory agency to defend the **reasonableness** of regulatory actions. This type of ambient biological response data can be particularly valuable in fulfilling that need.

The preceding discussion of the analysis of the capabilities of biological survey data to discriminate different types of impacts is contrary to several of the assertions of Suter (1993) in a critique of community indices. Although Suter (1993) made a number of thoughtful criticisms, the ones most relevant to the concept of Biological Response Signatures and the utility of the IBI and ICI include: (1) the IBI is composed of heterogeneous variables; (2) indices such as the IBI are too ambiguous to determine why an index value is high or low; (3) the inherent likelihood of eclipsing between different metrics (i.e., metrics cancel each other); and, (4) the IBI has no diagnostic power. The IBI comprises homogeneous variables as long as characteristics from a common organizational level such as fish, birds, chemicals, etc. are used. Many of Suter's concerns are based on the premise that they are heterogenous. There is merit to Suter's concern that an arbitrary mixing of variables, without any thought given to unintentional introductions of bias, compounding, and variance, is to be avoided. However, the charge that the IBI (and similar multimetric indices) are too ambiguous to determine why the index is high or low is refuted by the discussion of the Biological Response Signatures and the conceptual model of community response (Figures 13 and 14).

While we agree that an index value alone is almost meaningless, the information that results in an index value is obtained within the workings of a multidisciplinary framework where source, land use, and other chemical/physical water-related information are gathered and interpreted at the same time (e.g., lines of evidence within a causal associations framework). The chance that significant eclipsing will occur between different metrics is minimized and virtually eliminated by the careful construction of the IBI as modified by Ohio EPA (Ohio EPA 1987b). The example cited by Suter (1993) regarding the potentially eclipsing effect of organism density (numbers) and the frequency of diseases simply does not happen because of the use of only certain types of anomalies (e.g., the DELT anomalies) and the exclusion of highly tolerant species from the number of individuals metric. This latter modification prevents the mere increase in organism density from automatically producing a higher IBI value. The three-dimensional plots of the IBI, MIwb, and percent DELT anomalies also lend insight into and challenge the validity of Suter's concerns. The MIwb, comprising density and biomass components is **lowest** when the percent DELT is the **highest**, the antithesis of the eclipsing concern (Figure 7). This was also demonstrated in Figures 8 and 9 where organism density (including tolerants) was highest for the impact types (i.e., Municipal Conventional, Channelization, and Agricultural NPS) that exhibited much lower percent DELT anomalies than the Complex Toxic impact type, which generally exhibited the lowest density. A problem with the IBI occurs when organism densities are extremely low — the traditional scoring presumptions break down, particularly for the proportional metrics. To counter this problem we have devised a low-end scoring procedure (Ohio EPA 1987b). Thus, if the proper construction of the IBI (or similar evaluations) takes place up front, the eclipsing and other problems can be abated.

The most troubling of Suter's assertions is that the IBI has no diagnostic power. Over the linear scale of the IBI even gross changes have meaning (see Figure 13) with regard to the relative condition of the community and the types of impacts that are present. However, an IBI is not produced merely for the sake of generating a single-number-based evaluation. It is inherent in the IBI/regional reference site framework to simultaneously examine the aggregated and even the individual components in accomplishing an assessment of a particular waterbody. The use of key community attributes and metrics to examine if discernable patterns and combinations of responses exist for different types of cultural impacts (Figures 7 to 11) demonstrates that **some** level of diagnosis is possible. The Biological Response Signatures in combination with water quality, effluent, and habitat data was useful in demonstrating that the continuing impairment in the Ottawa River (Figures 4 and 5) was due to toxic impacts as opposed to habitat, the latter position taken by the regulated entities in an appeal of the use designation revision to WWH (see Chapter 9). While the application of this technology may not be entirely conclusive in every situation, it is informative and can lead to the cost-effective application of more powerful and intensive diagnostic tools (e.g., toxicity tests, chemical evaluation, and biomarkers). If the IBI is not used in this context then many of Suter's concerns will most likely retain their original validity.

5.0 SUMMARY

The preceding results demonstrate that discernable patterns in biological community information do exist and can be used to determine the probable cause of certain types of impairments, especially in combination with other source and instream chemical/physical data. This was best demonstrated for the Complex Toxic impact type, particularly for specific community indicators such as the percent DELT anomalies for fish and percent *Cricotopus* spp. for macroinvertebrates. The ability of these attributes **alone** was further improved when additional community indices, metrics, and other aggregations were included. The ability to discriminate between the other impact types varied widely with the ability to identify the CSO/Toxic impact type second only to the Complex Toxic type. Some attributes (number of sunfish species, percent caddisflies, and qualitative EPT) indicated the Channelization impact type reasonably well. However, there was a much broader overlap between the other impact types. Some of this is to be expected and is a result of the similarity of the environmental processes involved in each. For example, the Municipal Conventional, CSO/Urban, and Agricultural NPS impact types overlapped to a greater extent, which is not surprising since each involves the effects of nutrient enrichment. Examining finer aggregations of the community data than were done in this analysis could potentially increase the discriminatory power of the information to better separate these impact types.

These results point to the need to carry out biological monitoring to a prescribed level of effort, particularly in two areas: (1) taxonomic detail, and (2) the inclusion of multiple organism groups. With regard to taxonomy it was essential in our analysis to have the macroinvertebrates identified to the lowest possible level, particularly the midges. Some protocols do not require genus/species-level identifications for certain groups, which would have rendered the macroinvertebrate data much less useful in the analyses. For example, we would not have identified the utility of the genus *Cricotopus* in discerning the Complex Toxic impact type. National biological monitoring frameworks are currently debating this issue and should consider the future consequences of such choices. One possible conclusion from our analyses and the biocriteria framework (see Chapter 9) in general is that the increased "data dimensions" afforded by the more detailed taxonomy translates into more powerful and sensitive analytical tools such as the biological response signatures.

The inclusion of multiple organism groups also has significant benefits as well. The fish and macroinvertebrate assemblages each seem particularly well suited to defining different impact types and in combination further enhances this discrimination. For example, the fish community seems particularly well suited to identifying impairments due to macroscale influences such as habitat modification (i.e., Channelization and Impoundment impact types). In these situations the macroinvertebrates provide more specific information about water column effects as additional potential causes of impairment. Macroinvertebrates have been particularly responsive to indicating the severity of the CSO/Urban impact type. In addition, each group has different rates of recovery, the macroinvertebrates generally being quicker to respond over time than fish, but this has not been invariable. Eagleson et al. (1990), using a single indicator group (macroinvertebrates), cautioned about interpretations that could be confounded by seasonal variations and periodic absences of key indicator taxa. While we compensate for this potential problem via a standardized index sampling period, the inclusion of fish, which inhabit the receiving water year-round, can help to minimize this potential problem.

The Area of Degradation Value (ADV) provides a new tool to enhance the analysis of multimetric information beyond the traditional pass/fail assessments. By providing a quantified analysis of departures from a target index value (usually the ecoregional biocriterion) along a longitudinal continuum, both the severity and extent of an impairment can be evaluated. This allows for the analysis of incremental changes in aquatic community performance over space and time. As such, this has been a useful tool for demonstrating the effectiveness of pollution control efforts over time, and in comparing the severity of impairments between different stream and river segments. Future work is needed, however, to better utilize the ADV as part of resource value assessment such as NRDAs and in assessing other types of environmental damage claims.

ADV and Biological Response Signatures represent new tools that can be used as a key part of larger assessment efforts within which multiple goals are being pursued. For states, this would most commonly entail trend assessments and monitoring in support of various water quality management and regulatory programs such as water quality standards, NPDES permitting, and nonpoint source management and assessment. Clearly, other chemical, toxicological, physical, and source information (e.g., toxicity test results, pollutant loadings, permit violations, spills, etc.) must be used as part of the overall assessment process. The overall effort should focus on building improved "lines of evidence" in which the aquatic community response information is integrated with stressor and exposure indicator information to produce more accurate conclusions about likely causes and sources of observed biological impairments than would otherwise be accomplished relying on any single indicator class alone.

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